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Estimating Watershed Biodiversity: An Empirical Study of the Chesapeake Bay in Maryland, USA

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ABSTRACT There has been increasing demand for rigorous methods for evaluating biodiversity, one of the ecosystem services that sustains and fulfills human life. After carefully examining the literature, we found three key points that should be taken into account when we evaluate biodiversity. The first point is that any “indicator species” tends to be a leaky target of biodiversity. The second point is that “buffering” that is useful for representing the ecological concept of boundaries should have scientific meanings. The third point is that a “watershed” that integrates most natural processes is advantageous as the spatial range for evaluation. Based on these considerations, we developed the Biodiversity Probability Index (BPI) to account for biodiversity. We analyzed the relationship between BPI and total economic value of ecosystem services per unit area in order to test the reliability and utility of BPI. All BPI values were under a positive regression curve. The curve showed the highest economic value that could be achieved at a given biodiversity status. Based on our findings along the analysis, we conclude that the BPI has two advantages: (1) The BPI identifies the extent of the impact on biodiversity within a given landscape, and its composition of each category of species helps to identify what kind of wild life habitat could be seriously threatened by future land use development; (2) The BPI tells not only the biodiversity condition in ecological sense, but also provides the potential total economic value of ecosystem services, which will be achieved under ideal conditions.

KEYWORDS: ecosystem services, Biodiversity Probability Index, contagion effect, Habitat Unit, habitat requirement.

1. Introduction

Since the Convention on Biological Diversity went into effect in 1993, there has been increasing international demand for rigorous methods for evaluating biodiversity—one of the ecosystem services that sustains and fulfills human life. Currently, the evaluation of biodiversity is carried out in a somewhat haphazard and idiosyncratic fashion, with individual researchers and decision makers evaluating biodiversity in ways that satisfy their own research purposes and fit within the specific context of their chosen study site: i.e., assessing land use conversion and development impacts on wildlife habitat, prioritizing the selection of identified bio-reserves, and assessing mitigation effects on local communities (Lenders et al. 2001; Hermy et al. 2000; He et al. 2000; Bryja 2000; Duelli 1997; Duelli et al. 2001; Schwab et al. 2002). It often happens that the measurement of biodiversity is tailored to suit specific management goals, local environmental context and governmental policies. However, for the purpose of scientific generalizability and political consistency, we argue that it would be useful if a common method for biodiversity evaluation could be identified and applied consistently in different ecological and political contexts. After carefully examining the literature, we have delineated three crosscutting themes that should be taken into account in evaluating biodiversity, especially for assessing the effect of land use conversion and development impact.

First, there is the issue of *identifying biodiverse areas through indicator uses*. One of the most common approaches for selecting areas to represent regional biodiversity is to use “indicator” groups, such as rare species (Wu et al. 2000; Noss 1999). For example, the presence/absence of rare species has been incorporated into a green infrastructure GIS model, developed by the Maryland Department of Natural Resources, which helps decision makers estimate the potential ecosystem health of selected landscapes in the state (Weber et al. 2000). In the 1997 environmental impact assessment law proclaimed in Japan, several species that represent “superiority”, “typicality” and “specialty” of ecosystems were selected as targets for further investigation prior to land development. In such cases, a set of areas that is determined to contain all indicator taxa is generally assumed to be biodiverse. One limitation of this approach is that the set of species-rich areas for any given indicator group is likely to represent many other organisms only if the members of the group span a wide range of habitats or environment (Lenders et al. 2001; Taggart 1994). However, even if these selected species span a wide range, they will never cover all the acts of other species. In addition to this essential problem, collecting reliable species distribution data is a very time-consuming task. Therefore, it is necessary, for taking account of biodiversity, to find an approach other than a method using “indicator species”.

Secondly, there is the issue of *meaningful buffering*. With the widespread use of GIS technology, buffering has become one of the strongest techniques available for delimiting biodiverse regions; yet it is also a tool that can be used carelessly. Buffering is useful primarily for representing the ecological concept of boundaries that separate ecosystems or land uses within a given landscape. Landscape boundaries exhibit five basic ecological characteristics: *habitat*, *filter*, *conduit*, *source*, and *sink* (Forman et al. 1999, pp. 96), each of which relates to biodiversity (Harris 1984; Yahner 1988). The problem is that different species have vastly different effective “edge” widths. Thus, an appropriate buffer for one species may be insufficient for another species or too much for another (Schwartz 1999). Therefore, buffering distances need to be determined by scientific analysis and applied with care.

Finally, there is the issue of *spatial range for evaluation*. Increasingly, the importance of watershed management is being recognized, though other spatial ranges characterized by political and legal boundaries are commonly used in biodiversity evaluation as well (Lenders et al. 2001; Hermy et al. 2000). Watershed-based analysis is advantageous because it considers all activities within a landscape that affect watershed health. Furthermore, it integrates biology, chemistry, economics, and social considerations into decision-making (US Environmental Protection Agency Office of Water 2001). Based on these considerations, in this paper, we present a simple model for evaluating biodiversity. The Biodiversity Probability Index (BPI) is proposed and tested for its sensitivity to total economic value of ecosystem services in the Chesapeake Bay in Maryland, USA.

2. Methods

2.1 Site, Database and GIS software for analysis

Two hierarchical sets of watershed data for the state of Maryland, USA were acquired from the Maryland Department of Natural Resources. One data file delineates watersheds with a 6-digit identification number, thus representing the 19 largest river systems within the state. These 6-digit watershed areas

range from $13.7 \times 10^6 \text{ km}^2$ to $2.9 \times 10^9 \text{ km}^2$. Yet another file reveals 8-digit identification numbers for watersheds, which include nested information for the 6-digit watersheds.

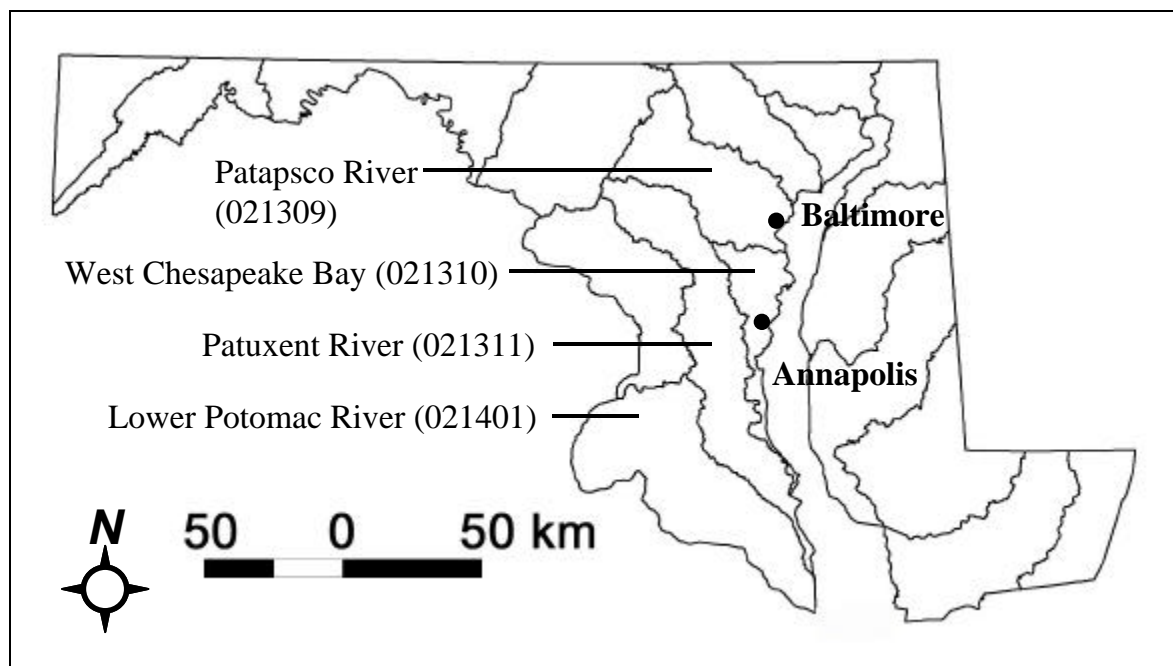


Figure 1. Four target watersheds with 6 digit-numbers in Maryland

The research team targeted the four watersheds of Patapsco River (021309), West Chesapeake Bay (021310), Patuxent River (021311), and Lower Potomac River (021401), and each of the eight sub-watersheds of Patapsco River and Patuxent River, in Maryland, USA (Figure 1). These watersheds differ greatly in their level of urbanization. For example, the Back River basin, which is one of the sub-watersheds of Patapsco River, is the most urbanized (77.1 percent) watershed in Maryland, and supports 1,285 people per square kilometers. On the other hand, in the lower basin of Patuxent River, only 15.1 percent of the land area is urbanized and the population density is 178 per square kilometers.

The research team used the Maryland portion of the National Land Cover Data set (USGS NLCD, Version 99-05 Edits) developed from LANDSAT-TM data acquired by the Multi-resolution Land Characterization Consortium. In order to fill data gaps, NLCD was overlaid with the Maryland Department Planning land use coverage database using Arc/Info 7.1. This grid overlay was converted into vector coverage (Land cover base map I) for making buffers and reclassifying them on ArcGIS™ as mentioned later.

2.2 Development of the Biodiversity Probability Index

2.2.1 Category of species type for evaluation

In the biodiversity literature, many different category types have been used for the assessment of species diversity, including: ecological community, species assembly type or conservation criteria regarding rarity and diversity (Peterken 1968; McKenzie et al. 1989; Margules et al. 1981). In such studies, actual species distribution lists are needed to effectively determine the level of biodiversity in a given area. The biggest limitation is that formal species distribution lists are very costly and time-consuming to collect. On the other hand, the criteria of environmental variation such as temperature, soil moisture and light intensity linked to species distribution, can be widely generalized, thus leading to their usefulness as an effective surrogate in the absence of biotic data (Faith et al. 1996).

Using proxy abiotic data, individual species can be categorized according to their habitat requirements: interior species, edge species and multihabitat species (Forman 1999, pp.96-97). These categories are usually independent of taxonomic group. For example, edge species live primarily near the border of a given spatial element in the landscape, interior species are primarily distant from a border, and multihabitat species are those requiring or using two or more habitat types. These category types can be applied to non-aquatic areas: forest, grassland, and wetland. In this study, six categories of species based on habitat requirement are identified: *forest species*, *grassland species*, *wetland species*,

edge species, aquatic species, and multihabitat species.

2.2.2 Edge, interior and transition zones

Sun and wind exhibit overriding controls on edge microclimate in a regional scale of the Chesapeake Bay. They have an effect on the edge microclimate, such as CO₂ and H₂O content of the air as a result of photosynthesis and respiration of soils (Kapos et al. 1993), as well as habit of insects and other animals in the edge areas (Gates et al. 1996; Bushman et al. 1988; Gates et al. 1996). These ecological effects increase with a greater difference in vegetation height between the adjacent ecosystems. Figure 2 shows the spatial relationship of interior, edge, and transition zones based on abiotic controls on microclimate (Geiger 1965; Heisler et al. 1988; Chen et al. 1992) and various forest interior perceptions by plants and animals (Kapos et al. 1993; Bushman et al. 1988) are shown.

As Figure 2 exhibits, a forest darkens the field edge; its width based on illumination being approximately equal to the height of the trees. In contrast, the presence of a field lightens the forest, elevated levels extending a few meters into the forest (Forman et al. 1999). On the other hand, air flowing over a field is typically reduced in velocity to a distance of $8h$ (h = height of canopy; at upwind of a forest) to $25h$ (at downwind of the forest) (Heisler et al. 1988). In addition, wind penetration into the forest is about $0.5h$ (at upwind of a forest) to $1h$ (at downwind of a forest) (Chen et al. 1992).

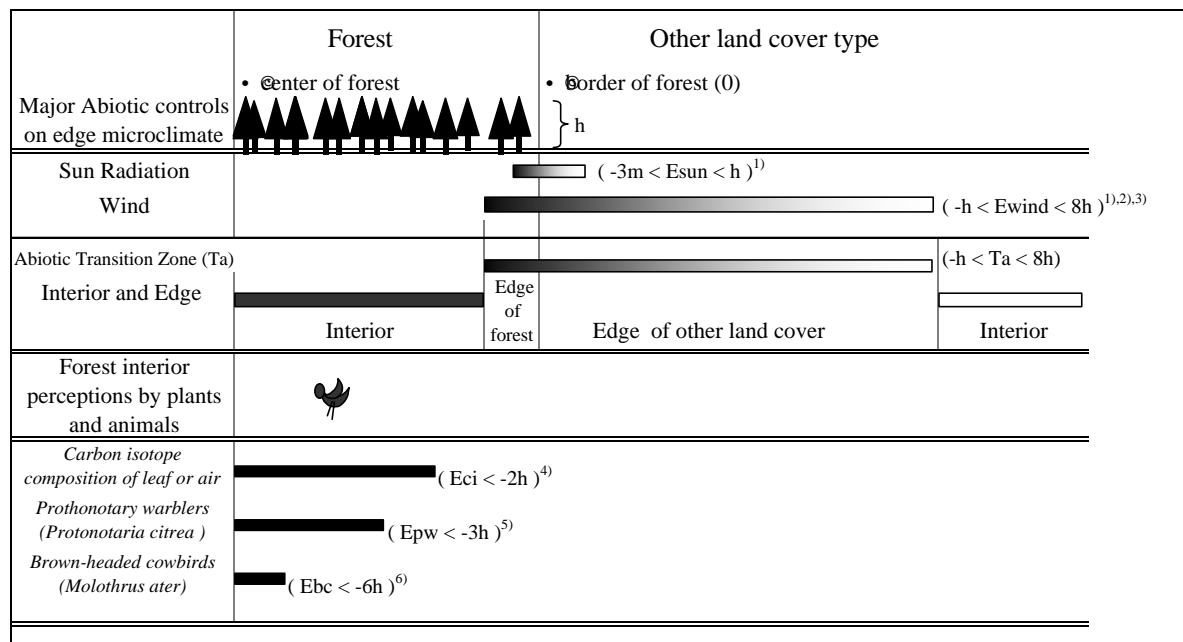


Figure 2. Spatial relationship of interior, edge, and transition zones based on abiotic controls on microclimate and various forest interior perceptions by plants and animals.

h = canopy height; Deciduous forest = 40m, Evergreen forest = 25m, Mixed forests = 35m, Wetland forest = 25m. E_{sun}: Edge range controlled by sun radiation; E_{wind}: Edge range controlled by wind; E_{ci}: Edge range for trees; E_{pw}: Edge range for Prothonotary warblers; E_{bc}: Edge range for Brown-headed cowbirds.

1) Geiger 1965, 2) Heisler et al. 1988, 3) Chen et al. 1992, 4) Kapos et al. 1993, 5) Bushman et al. 1988, 6) Gates et al. 1996.

Using this information, we can estimate an abiotic transition zone beginning from at least $0.5h$ inward from a border of forest and extending no less than $8h$ outward from a border of forest. The rest of forest and field are forest interior and field interior.

2.2.3 Reclassification of land cover data

Based on the abiotic boundary concept, we reclassified our land cover base map I in ArcGIS™ version 8.2. Canopy heights were estimated from various external sources of information about canopy height in Maryland: canopy height data acquired from satellite images (Harding et al. 2001; Lefsky et al. 1999) and standard heights of tree species listed in botanical books (Little 1980).

Deciduous or mixed forest dominated by *Liriodendron tulipifera* is the most common upland

forest in Maryland. Experimental data available for this type of mature stand shows that maximum canopy height is about 40m in a deciduous forest (Harding et al. 2001) and 35m in a mixed forest (Lefsky et al. 1999). Standard heights of common evergreen and wetland tree species in Maryland are shown in Table 1. Fourteen species of evergreen trees are commonly grown (Little 1980), while in wetlands, 8 species are known to frequently occur (more than 66% occurrence in wetland) (Maryland Department of the Environment 2002). The average height for both evergreen and wetland trees is about 25m.

Table 1. Standard heights of common tree species in Maryland

Evergreen tree species	Height* (m)
<i>Chamaecyparis thyoides</i>	27
<i>Juniperus virginiana</i>	24
<i>Pinus echinata</i>	30
<i>Pinus rigida</i>	21
<i>Pinus taeda</i>	30
<i>Pinus virginiana</i>	18
<i>Picea abies</i>	24
<i>Picea rubens</i>	24
<i>Pinus palustris</i>	30
<i>Pinus pungens</i>	12
<i>Pinus resinosa</i>	24
<i>Pinus serotina</i>	21
<i>Pinus strobus</i>	33
<i>Tsuga canadensis</i>	21
Wetland tree species	Height* (m)
<i>Acer saccharinum</i>	24
<i>Betula nigra</i>	24
<i>Fraxinus pennsylvanica</i>	18
<i>Platanus occidentalis</i>	30
<i>Quercus bicolor</i>	21
<i>Quercus palustris</i>	27
<i>Salix nigra</i>	30
<i>Taxodium distichum</i>	37

*The heights are taken from the description in botanical books (Little, 1980)

Based on this data, we set the canopy height for deciduous forest at 40m, evergreen forest at 25m, mixed forest at 35m, and wetland forest at 25m. We reclassified “edges” of developed land, grasslands, forest, wetlands, and barren land by using the buffering command on base map I. Eleven primary groups resulted in a new land cover base map II: *water, developed land, edge of developed land, forest, edge of forest, grasslands, edge of grasslands, wetlands, edge of wetlands, barren land, and edge of barren land*. Figure 3 shows a part of land cover base map I and the corresponding part of land cover base map II. Following the reclassification procedure, base map II was converted into 30 m grids for the estimation of Biodiversity Probability Index. The grid size was determined so as to be the same resolution of the original USGS NLCD.

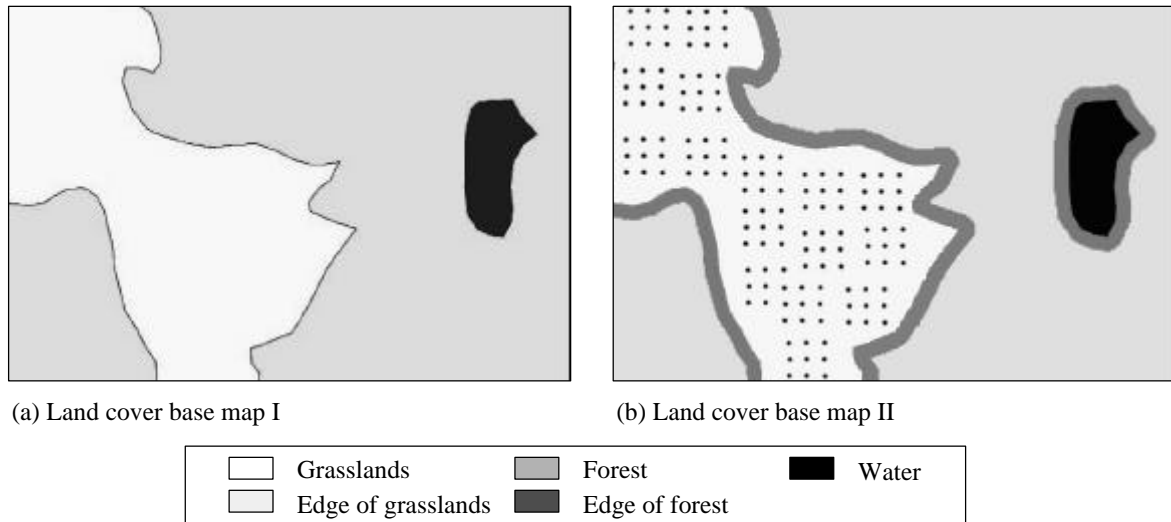


Figure 3. Effect of buffering based on abiotic controls on microclimate

2.2.4 Biodiversity Probability Index

Habitat Evaluation Procedure (HEP) is one of the most commonly used ways of assessing the effect of land use development on local biodiversity in the United States. In general HEP, HU (Habitat Unit), habitat value (vary from 0 to 1) for selected species multiplied by habitat land area, is calculated as the index of biodiversity. This procedure has a great advantage to be clear and logical though it is lack of spatial patterns of landscape. We developed Biodiversity Probability Index by improving the HEP method.

The Biodiversity Probability Index (BPI) for each watershed represents the sum total HU (Habitat Unit) for *forest species*, *grassland species*, *wetland species*, *edge species*, and *multihabitat species* per unit area (here, per square kilometer). Habitat Unit is estimated by the habitat value (V), land area within a watershed (A) and contagion index (Q).

$$BPI = S (HU_j) / A \quad (1)$$

$$HU_j = S (Q_i * A_i * V_{ij}) \quad (2)$$

$$Q_i = N_{i,i} / N_i \quad (3)$$

Where

i : Land cover type (*Water, Developed land, Edge of developed land, Forest, Edge of forest, Grasslands, Edge of grasslands, Wetlands, Edge of wetlands, Barren land, and Edge of barren land*)

j : Species category (*Forest, Grassland, Wetland, Edge, and Multihabitat species*)

HU_j : Habitat unit of species category j

Q_i : Contagion index of i

A_i : Land area of land cover type i

V_{ij} : Habitat value of land cover type i for species category j

$N_{i,i}$: Number of cells of i adjacent to i

N_i : Total number of i

Table 2 shows habitat value (V) of land cover for each category of species. A habitat value of “0” means no chance of being present, and “1” means some chance of being present. Land area represents local significance, while contagion effect expresses landscape context. Contagion effect is adapted here to take into account the knowledge that population growth and survival rate tend to be greater in a patch connected with other patches than in an isolated patch (see Forman 1999, pp. 275), which is represented as the probability of cells of land use type i being adjacent to cells of land use type i (O’Neil et al. 1988).

BPI can thus be interpreted as representing the *potential of encountering any one of the five species types* within a given watershed.

Table 2. Habitat value (V) for each category of species

CODE	Land cover type	Habitat value (V)					
		Forest species	Grassland species	Wetland species	Edge species	Aquatic species	Multihabitat species
100	Developed land	0	0	0	0	0	1
110	Edge of developed land	0	0	0	1	0	1
200	Grasslands	0	1	0	0	0	1
210	Edge of grasslands	0	0	0	1	0	1
400	Forest	1	0	0	0	0	1
410	Edge of forest	0	0	0	1	0	1
500	Water	0	0	0	0	1	1
600	Wetlands	0	0	1	0	0	1
610	Edge of wetlands	0	0	0	1	0	1
700	Barren land	0	0	0	0	0	1
710	Edge of barren land	0	0	0	1	0	1

Ranking: 0=None, 1=Presence

2.2.5 Estimation of economic value of ecosystem services

Ecosystem services consist of flows of materials, energy, and information from natural capital stocks, which combine with manufactured and human capital services to produce human welfare. In order to evaluate the economic value of ecosystem services, both market and non-market components of the value of ecosystem services have been estimated for each biome (Costanza et al. 1997). In this paper they have estimated seventeen ecosystem services for sixteen land cover types, based on published studies and a few original calculations. Using the unit value of each land cover type, total economic value of ecosystem services (TEV) in each watershed was estimated.

The unit values of each land cover type used in our calculations are: “Water” = \$ 2,243,065 km⁻² year⁻¹ (Estimated from the unit value for “Lakes/rivers” and “Estuary” from the estimates by Costanza et al. (1997), explained after), “Developed land” = \$ 0 km⁻² year⁻¹ (Adopted from “Urban”), “Grasslands” = \$ 9,200 km⁻² year⁻¹ (Adopted from “Cropland”), “Barren land” = \$ 0 km⁻² year⁻¹ (Adopted from “Ice/rock”), “Forest” = \$ 30,200 km⁻² year⁻¹ (Adopted from “Temperate/boreal”), “Wetlands” = \$ 1,478,500 km⁻² year⁻¹ (Adopted from “Wetlands”). “Water” included both fresh water of rivers and lakes (3 %) and estuaries (97 %) in land cover base map II. Thus, the unit value of “Water” was estimated by taking a weighted average of the unit value of “Lakes/rivers” (\$ 8,498 km⁻² year⁻¹) and “Estuary” (\$ 2,283,200 km⁻² year⁻¹) from the estimates by Costanza et al. (1997). [Namely, \$ 8,498 x 0.03 + \$ 2,283,200 x 0.97 = \$ 2,243,065]

2.2.6 Comparison of BPI with other indices

In order to test the reliability and utility of BPI, we analyzed the relationship between BPI and total economic value of ecosystem services per unit area, and also analyzed the relationships between other indices developed by Maryland Department of Natural Resources and total economic value of ecosystem services per unit area. Four indices that evaluate wildlife status of sub-watersheds of Patapsco River (021309) and Patuxent River (021311), which are open to the public at the web site of Maryland DNR, were selected, i.e., *Imperiled Aquatic Species Index*, *Non-tidal Benthic Index of Biotic Integrity*, *Designated Wildland Area* and *Water Quality Habitat Index* (refer to the Glossaries for these definitions). These are common approaches used in the evaluation of habitat integrity. The former two indices are indicator species approaches, and the latter two indices are environmental quality approaches.

3. Results and Discussion

3.1 Evaluation of watersheds by Biodiversity Probability Index

Table 3 shows land area and *Contagion Index* (CI) of each land cover type and *Biodiversity Probability Index* (BPI) of four watersheds given 6-digit numbers by Maryland DNR: Patapsco River (021309), West Chesapeake Bay (021310), Patuxent River (021311), and Lower Potomac River (021401).

Table 3. Land area and Contagion Index (CI) of each land cover type and BPI for 6-digit watersheds: Patapsco River (021309), West Chesapeake Bay (021310), Patuxent River (021311) and Lower Potomac River (021401).

CODE	6-digit Number	021309		021310		021311		021401	
	Land cover type	Area (km ²)	CI	Area (km ²)	CI	Area (km ²)	CI	Area (km ²)	CI
100	Developed land	339.7	0.86	76.8	0.67	177.6	0.69	102.0	0.63
	Edge of developed								
110	land	477.9	0.70	194.3	0.66	442.1	0.66	181.9	0.59
200	Grasslands	57.5	0.62	20.0	0.54	164.8	0.63	212.4	0.66
210	Edge of grasslands	286.6	0.66	81.5	0.61	481.7	0.66	246.3	0.61
400	Forest	310.9	0.61	221.1	0.69	719.0	0.66	847.7	0.73
410	Edge of forest	157.6	0.01	90.5	0.01	300.4	0.01	262.1	0.01
500	Water	116.6	0.83	96.8	0.76	150.2	0.83	1034.7	0.96
600	Wetlands	2.4	0.51	4.6	0.51	17.7	0.59	18.3	0.55
610	Edge of wetlands	4.1	0.53	3.7	0.44	13.4	0.50	14.0	0.49
700	Barren land	1.3	0.60	0.2	0.07	1.1	0.54	1.5	0.45
710	Edge of barren land	4.7	0.53	0.8	0.44	10.0	0.59	3.3	0.52
	BPI	1.138		1.128		1.131		1.413	

The land areas and CI of grasslands, forest, and water in the Lower Potomac River watershed (021401) were the biggest among the four study watersheds, and contributed the highest BPI in all. On the other hand, in the watershed of West Chesapeake Bay (021310), land area of grassland, forest, and water were the smallest among the four watersheds, and their CI were relatively small among the four watersheds, and resulted in the lowest BPI overall.

As the table also shows, the Patapsco River watershed (021309) contained the largest amount of developed land and its CI was the highest among the four watersheds, which indicates that Patapsco River watershed is the most intensely urbanized among the four watersheds. Although the area of developed land in Patuxent River watershed (021311) was half the size of that in Patapsco River watershed (021309), the area of edge of developed land in Patuxent River watershed was almost the same as that in Patapsco River watershed (021309), which means that more developed land in Patuxent River watershed (021311) verges on forests than in the Patapsco River watershed. High concentration of urbanized area seemed to make the biodiversity in Patapsco River watershed high as that in Patuxent River watershed.

Figure 4 shows the composition of each category of species that contribute to BPI in each watershed. *Multihabitat species* occupied the biggest portion (more than 50%) among the six categories of species in all watersheds. *Multihabitat species* becomes the main component that contributes to BPI because of the Habitat Value identification (Table 2). However, percentage of other category of species shows the uniqueness of each watershed. In the most urbanized Patapsco River watershed (021309), *edge species* ranked second (27%). The Lower Potomac River (021401) stood out in four watersheds for its large contribution of *aquatic species* (24%).

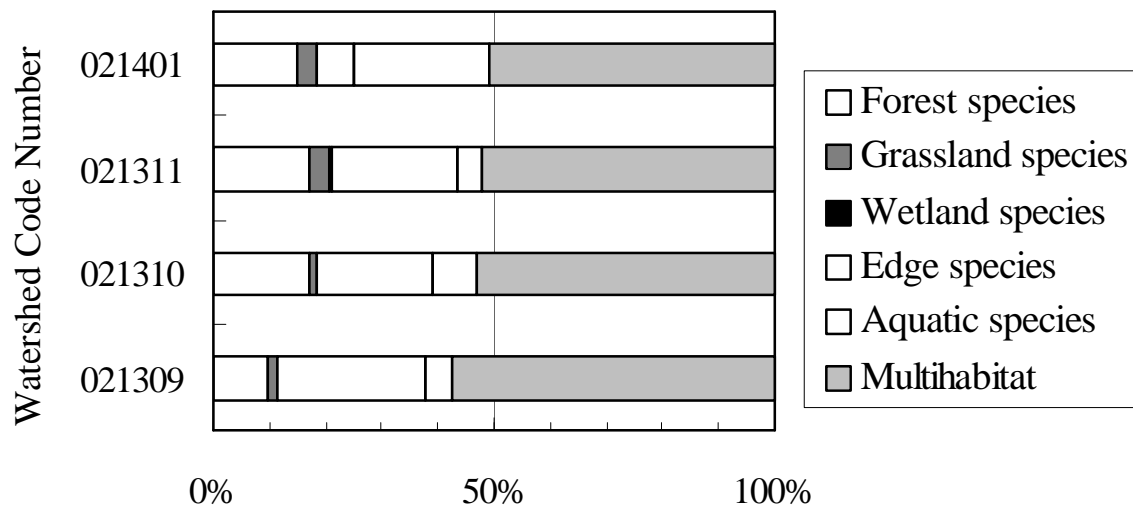


Figure 4. Species structure of BPI in each watershed.

3.2 Contribution of biodiversity to total economic value of ecosystem services in watershed

DeGroot et al. (2002) grouped ecosystem functions into 4 primary categories, *regulation*, *habitat*, *production*, and *information functions*, so that associated ecosystem services could be analytically distinguished from one another. According to this typology, biodiversity is one of the ecosystem services derived from “habitat functions” of ecosystems. However, biodiversity is not only one of the ecosystem services but also serves as the basis for many other ecosystem services (DeGroot et al. 2002). Thus biodiversity should have great influences on economic values of total ecosystem services. In this context, we analyzed how biodiversity contributes to total economic value of ecosystem services in order to test the reliability and utility of BPI. The relationship between BPI and total economic value of ecosystem services per unit area is shown in Figure 5. The marks at low BPI varied widely from high to low; however, all marks were under a positive regression curve of some BPI and economic value ($r^2 = 0.902$). In other words, the curve shows the highest economic value that could be achieved at given biodiversity status. Lower economic values in some watersheds indicate that other ecosystem services than biodiversity does not exercise their full capacity for bringing out those economic values. We will be able to identify what kind of ecosystem services are controlling the total economic value by analyzing the relationships between other ecosystem services than biodiversity and total economic value of ecosystem services.

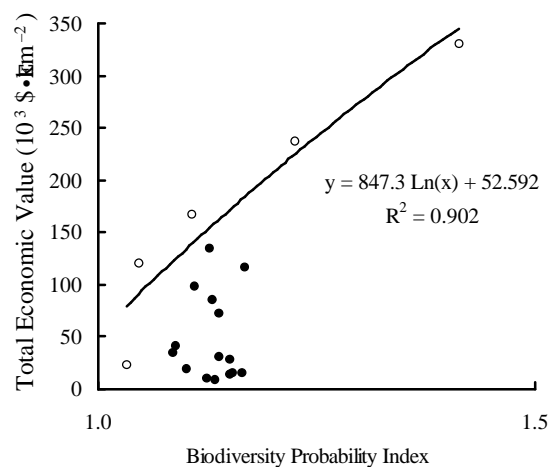


Figure 5. The relationship between BPI and total economic value of ecosystem services in watersheds.
 ? : data used for estimation of regression curve, ○ : other data

Figure 6 shows the relationships between the other four indices developed by Maryland DNR and total economic value of ecosystem services per unit area. The number of plotted data are different among (a), (b), (c) and (d). This is because there is no area or species that satisfy the identification of some indices in some watersheds. No meaningful relationships were detected between them, contrary to the significant relationship between BPI and total economic value of ecosystem services per unit area (Figure 5). “Imperiled Aquatic Species Index”, “Non-tidal B-IBI”, and “Water Quality Habitat Index” put much emphasis on aquatic ecosystems, so they may not show good correlations with total economic value of ecosystem services. Meanwhile, “Designated Wildland Area” places value on unique ecosystems; therefore, it misses taking the ordinary ecosystems into account. This biased evaluation might be the reason why this index showed mal-correlation to total economic value of ecosystem services per unit area. It is obvious that BPI has much stronger reliability and utility than these four indices for the evaluation of biodiversity.

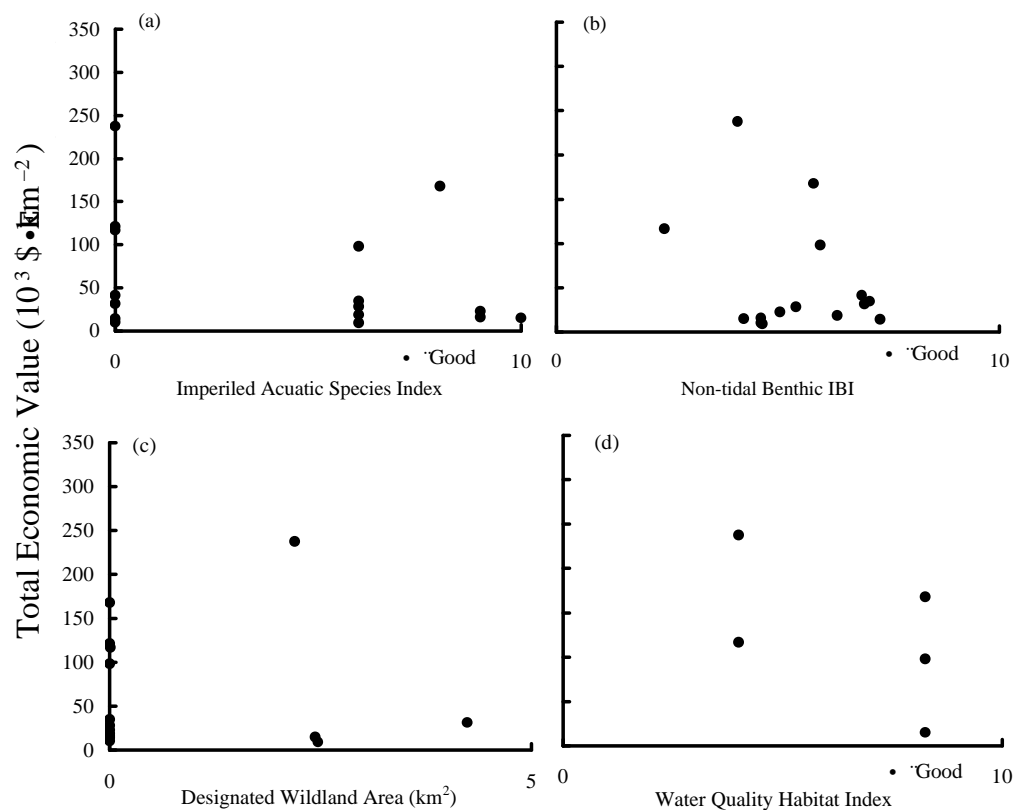


Figure 6. The relationships between (a) Imperiled Aquatic Species Index, (b) Non-tidal Benthic Index of Biotic Integrity, (c) Designated Wildland Area, (d) Water Quality Habitat Index, and total economic value of ecosystem services.

3.3 Advantages of Biodiversity Probability Index

At the outset of this paper, we pointed out three key points that should be taken into account when evaluating biodiversity, especially when assessing the effects of land use conversion and development impact. Our first observation was that any “indicator species” tends to be a leaky target of biodiversity. Thus, we aimed our analysis at the environmental variation linked to species distributions. The resulting BPI targets whole species - *forest species*, *grassland species*, *wetland species*, *edge species*, *aquatic species* and *multihabitat species* - associated with environmental criteria characterized by abiotic controls on microclimate. Our second observation was that “buffering” should have an underlying scientific meaning when and where it is applied. In this study, we investigated pertinent information on canopy height in Maryland and used it to create a buffer delineating the distance to landscape edges. Our third observation was that the “watershed” concepts integrate most natural processes and is therefore advantageous as a spatial boundary for biodiversity evaluation. BPI is flexible enough to suit a wide range of watersheds.

Based on our findings along the analysis using BPI, we conclude that it has two advantages: (1) The BPI identifies the extent of the impact on biodiversity within a given landscape, and its composition of each category of species helps to identify what kind of wildlife habitat could be seriously threatened by future land use development. (2) The BPI tells not only the biodiversity condition in an ecological sense, but also provides the potential total economic value of ecosystem services, which will be achieved under ideal conditions.

Biodiversity is a concept that is inherently difficult to represent as one sole index. Considering that many alternative methods and indexes have been developed (for example, Lenders et al. 2001; Hermy et al. 2000; He et al. 2000; Bryja 2000; Duelli 1997; Duelli et al. 2001; Schwab et al. 2002), it is challenging to form a good combination of these or to choose the most appropriate one that is suitable for local ecosystems. The usefulness and meanings of any model will not be clear until the relationships between the information from different theories have been closely examined, as done in this study. For example, Gustavson et al. (2002) examined the extent to which a biophysical-based index is proportionally reflected in monetary exchange, and found out the role of ecosystem is not being treated consistently. Exploring the juncture between an ecosystem-based application of information theory and economic information theory has potential fruitfulness. It is expected that the potential of BPI will be exercised more by investigation of its respect to other ecosystem services.

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Glossaries

Definitions of four indices below are reprinted from the web site "Watershed Profiles" by Maryland Department of Natural Resources (<http://mddnr.chesapeakebay.net/wsprofiles/surf/prof/prof.html>).

"Imperiled Aquatic Species Index": Maryland DNR's Wildlife and Heritage Division lists particular species, including some species of amphibians, fish, crayfish and mussels, as rare, threatened or endangered. To develop the index mapped in this indicator, distributions of these aquatic animals within the 8-digit watersheds were determined using Maryland Biological Stream Survey data and were scored from 0 - 10, based on the number of sites with rare species, their status (endangered, rare...), and the diversity of aquatic animals. The presence of rare, threatened or endangered species indicates the presence of suitable habitat, usually unmodified or minimally modified by human activity.

"Non-tidal Benthic Index of Biotic Integrity (B-IBI)": Benthic macroinvertebrates are used as biological indicators because they are reliable and sensitive indicators of habitat quality in aquatic environments. B-IBI compiles a number of measures into a single score of benthic community health. Included in calculating the B-IBI are measures of species diversity, species composition, productivity, and trophic

composition (Ranasinghe et al. 1997). For the purposes of the Unified Watershed Assessment, the original 1-to-5 scale was expanded to a scale of 1 (poor health) to 10 (good health) to be compatible with the other indices.

“Designated Wildland Area”: The area of wildland, which the Maryland General Assembly has formally designated to retain their wilderness character under the 1971 Maryland Wildlands Act. The law provides for three types of Wildland: one is a primitive area which by its size or location is, in effect, untouched by urban civilization; the second is a scientifically important unit, especially for ecology; the third is not of particular ecological or primitive stature but has the appearance of being in an untouched natural state, or could attain that appearance if held and managed for such a purpose. None of the presently designated Wildlands falls into this third type; most are of the second type, while six in Western Maryland are of the first.

“Water Quality Habitat Index”: It incorporates information on the status (1994-1996) for three parameters important in judging the habitat quality of estuarine waters: abundance of algae, water clarity, and dissolved oxygen levels. Abundance of algae is estimated using surface chlorophyll levels, water clarity is measured using secchi disk depths, and dissolved oxygen is measured in bottom waters during the summer months (July - September).

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